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Water and nitrogen processes along a typical water flowpath and streamwater exports from a forested catchment and changes after clear-cutting: a modelling study

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Abstract

A two dimensional model, FEMMA, to describe water and nitrogen (N) fluxes within and from a forested first-order catchment (Kangasvaara in Eastern Finland) was constructed by linking the most significant processes affecting the fluxes of water, ammonium, nitrate and dissolved organic nitrogen along a hillslope from the water divide to the stream. The hillslope represents the average flowpath of water in the catchment and the model was used to estimate the N fluxes for a catchment in eastern Finland before and after clear-cutting. The simulated results were in reasonable agreement with the nitrate, dissolved organic N and dissolved total N measurements from the study catchment and with other results in the literature. According to the simulations, the major sinks of N after clear-cutting were immobilisation by soil microbes, uptake by ground vegetation and sorption to soil. These sinks increased downslope from the clear-cut area, indicating the importance of an uncut buffer zone between the stream and the clear-cut area in reducing N exports. The buffer zone retained 76% of the N flux coming from the clear-cut area. Nitrification was a key process in controlling the N export after clear-cutting and N increases were mainly as nitrate. Most of the annual N export took place during the spring flood, when uptake of N by plants was minimal.

Keywords: ammonium, boreal, buffer zone, DON, FEMMA, nitrate, harvesting, watershed, catchment

Introduction

Nitrogen (N) cycling in pristine boreal forested catchments is tight, with only minor leaching losses (Tamm, 1991). Dissolved ammonium ($\text{NH}_4\text{-N}$), nitrate ($\text{NO}_3\text{-N}$) and organic N (DON) brought into soil water by atmospheric deposition or organic matter decomposition are efficiently taken up by trees, ground vegetation and microbes. The leaching of $\text{NH}_4\text{-N}$ and DON is reduced by sorption in the soil (Qualls, 2000). Nitrate, in contrast, is poorly retained by sorption and can leach to the catchment outlet. However, nitrification is small in boreal coniferous forest soils because of low pH and availability of $\text{NH}_4\text{-N}$ (Smolander *et al.*, 1995; Paavolainen and Smolander, 1998). In boreal forested catchments, water draining from upland mineral soils usually flows through a mire area before reaching the outlet. This affects stream

water quality and most dissolved N is in the form of DON (Lepistö *et al.*, 1995; Lamontagne *et al.*, 2000; Mattsson *et al.*, 2003).

Disturbances in the catchment, such as thorough clear-cutting and subsequent site preparation, fertilisation or increased N deposition, may increase dissolved N concentrations in the soil as well as N exports from the catchment (e.g. Grip, 1982; Bredemeier *et al.*, 1998; Ahtiainen and Huttunen, 1999; Pardo *et al.*, 2002). Increased N exports are detected mainly as elevated $\text{NO}_3\text{-N}$ levels in streams but elevated $\text{NH}_4\text{-N}$ levels have also been reported (Moldan and Wright, 1998a; Bäumlér and Zech, 1999). However, increases in DON due to clear-cutting or increased N depositions are considered not to occur (Akselsson *et al.*, 2004).

The time lag between the disturbance and the response in the stream varies according to catchment properties. In high N deposition areas or in catchments with fertile soil, stream water N concentrations increase rather quickly, whereas in low deposition areas or in nutrient-poor catchments, the response can be delayed and small (Bredemeier *et al.*, 1998). Although N export may increase many times after clear-cutting (Ahtiainen and Huttunen, 1999; Pardo *et al.*, 2002), it does not exceed the deposition inputs, except if the soils are N saturated (Akselsson *et al.*, 2004).

Increased N leaching after clear-cutting results from changes in N production and assimilation and hydrological fluxes; these changes are related to the amount of decomposable organic matter, disruption in uptake of N by trees, cessation of the interception of N by the forest canopy and changes in species composition of ground vegetation and microbial populations. Hydrological changes caused by clear-cutting include increased accumulation of snow, earlier onset of snowmelt and higher snowmelt intensity and elevated groundwater table levels (Päivänen, 1982; Troendle, 1983; Whitaker *et al.*, 2002) and increased runoff (Stednick, 1996). During the growing season both interception and transpiration decrease due to the removal of the tree stand (Calder, 1990; Buttle *et al.*, 2000; Whitaker *et al.*, 2002).

Clear-cutting in Fennoscandia has traditionally removed stem wood only and logging residues, left on site, containing hundreds of kg of N ha⁻¹ (Finér *et al.*, 2003) are subject to decomposition by soil fauna and microbes. Although clear-cutting may alter soil microbial communities (Bååth *et al.*, 1995; Houston *et al.*, 1998; Lindo and Visser, 2003), the consequences on net N mineralisation may be enhanced (Olsson *et al.*, 1996) or reduced (Bauhus, 1996). In a litterbag experiment, Palviainen *et al.* (2004) found significant immobilisation of N by logging residues with no net release of N during the first years after harvesting. Yet, other studies have found increased levels of dissolved N in soil water after harvesting (Rosén and Lundmark-Thelin, 1987; Titus and Malcom, 1992; Kubin, 1995; Piirainen *et al.*, 2002a).

Increases in NH₄-N availability after clear-cutting may activate nitrification in the soil (Paavolainen and Smolander, 1998; Smolander *et al.*, 1998). Nitrification is driven by the density and activity of the microbial community (Bengtsson *et al.*, 2003), and shows high spatial variability (Kjønaas *et al.*, 1998; Devito *et al.*, 1999). Increased NO₃-N availability and elevated groundwater table levels after clear-cutting can initiate denitrification (Paavolainen and Smolander, 1998) and N losses to the atmosphere.

After clear-cutting, ground vegetation can be a significant N sink (Likens *et al.*, 1970; Fahey *et al.*, 1991; Emmet

et al., 1991; Palviainen *et al.*, 2005) and conditions may favour grasses but suppress dwarf shrubs and mosses (Nykqvist, 1997). Species favoured after clear-cutting often have high N content, leading to an accumulation of N in the ground vegetation (Fahey *et al.*, 1991; Mou *et al.*, 1993; Palviainen *et al.*, 2005). However, in other studies, the logging residues have suppressed the ground vegetation (Nykqvist, 1971; Fahey *et al.*, 1991). Assimilation into soil microbes and chemical sorption by the soil has resulted in soil becoming a significant sink of N after timber harvesting (Kjønaas *et al.*, 1998; Lamontagne *et al.*, 2000; Vestgarden *et al.*, 2003).

During transport of dissolved N to the stream, different soil and vegetation types and microbial populations are encountered and all may control the form and levels of N reaching the stream. Clearly, the flow path is significant in controlling the N load to a stream (Devito *et al.*, 1999; Jacks and Norrström, 2004). The use of uncut buffer zones between a clear-cut area and receiving surface waters is based on the idea that denitrification and N retention by microbes, vegetation and soil occurs along the flow path taken. While buffer zones have been shown to decrease N exports to watercourses (e.g. Ahtiainen and Huttunen, 1999), there is only weak evidence of an increase in N uptake by trees in the buffer zone (Lundell and Albrektson, 1997). However, experiments in Sweden found retention of a significant proportion of N leached from a clear-cut area in a 10–30 m wide peatland buffer zone (Jacks and Norrström, 2004).

The export of N to streams is controlled by complex physical, chemical and biological interactions. Several models have been used to describe the water and N dynamics of a catchment, e.g. MAGIC and its versions (Jenkins *et al.*, 1997; Wright *et al.*, 1998a; Krám *et al.*, 2001; Jenkins *et al.*, 2001), SOILN (Eckersten *et al.*, 1995; Eckersten and Beier, 1998), SMART and its versions (Ahonen *et al.*, 1998; Kämäri *et al.*, 1998), MERLIN (Kjønaas and Wright, 1998; Wright *et al.*, 1998b) and pnet-CN/CHESS (Postek *et al.*, 1995). In these models, the water and N fluxes for a forest stand or an entire catchment are either calculated in a forested vertical column of soil divided into horizontal layers, or the modelling domain is described as a set of interconnected storages with no dimensions specified. For large catchments, various semi-distributed models have been applied, such as INCA (Whitehead *et al.*, 1998; Wade *et al.*, 2002; Langusch and Matzner, 2002), SWAT and related models (Krysanova *et al.*, 1998; Francos *et al.*, 2001) and HBV-N (Arheimer and Wittgren, 2002). In these models the focus has been on producing water and N dynamics for large river basins and process descriptions have been less detailed than in the point-scale models. In none of the approaches is the downslope routing of N through different

land-use types possible, therefore, they do not allow the assessment of how the location of a clear-cut area in the catchment affects the N export to the stream. Furthermore, none of the models accounts explicitly for fluxes of DON, even though DON can comprise 80–90% of stream export loads of Dissolved Total Nitrogen (DTN) in boreal forested catchments. In Finland and Sweden, an uncut buffer zone is required between a clear-cut area and the stream to reduce nutrient leaching. Clearly, the dimensions of the buffer zone must be optimised to minimise the loading to streams while maximising returns from harvesting.

To account for the most significant water and N fluxes in forested first-order catchments, a mathematical model (FEMMA) was constructed; this allowed evaluation of the effect of location of clear-cutting on stream water N export. The catchment was simplified into a two-dimensional hillslope extending from the water divide to the outlet. FEMMA was applied and parameterised for boreal nitrogen-limited, low N deposition conditions. In this study, FEMMA was applied to the Kangasvaara catchment for five years before and after clear-cutting to quantify the effects of clear-cutting on:

- (i) the N fluxes ($\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$ and DON) within forest, clear-cut and buffer zone compartments,

- (ii) N export to the stream, and
- (iii) to identify the dominant factors and processes.

By assessing mass balances for N fluxes in forest, clear-cut and buffer zone compartments along the hillslope, the significance of the buffer zone was evaluated.

Material and methods

STUDY AREA

Kangasvaara, a forested first-order, research catchment of 56 ha in Eastern Finland ($63^\circ 51' \text{ N}$, $28^\circ 58' \text{ E}$, Fig. 1), was studied intensively in an investigation of the effects of clear-cutting on water quality (Finér *et al.*, 1997). Elevations in the catchment range from 187 to 238 metres a.s.l. Long-term mean annual precipitation and air temperature in the area are 700 mm and 1.5°C , respectively. Most of the catchment (92%) is covered with upland mineral soil, while the rest is pristine forested peatland bordering the catchment stream. The mineral soil, mainly Haplic Podzols (FAO, 1988) is developed on sandy till between 0.9 to 2.1 m thickness with a fine earth clay content of less than 2%. The underlying bedrock is granodiorite and the site type is classified as medium rich *Vaccinium-Myrtillus* according

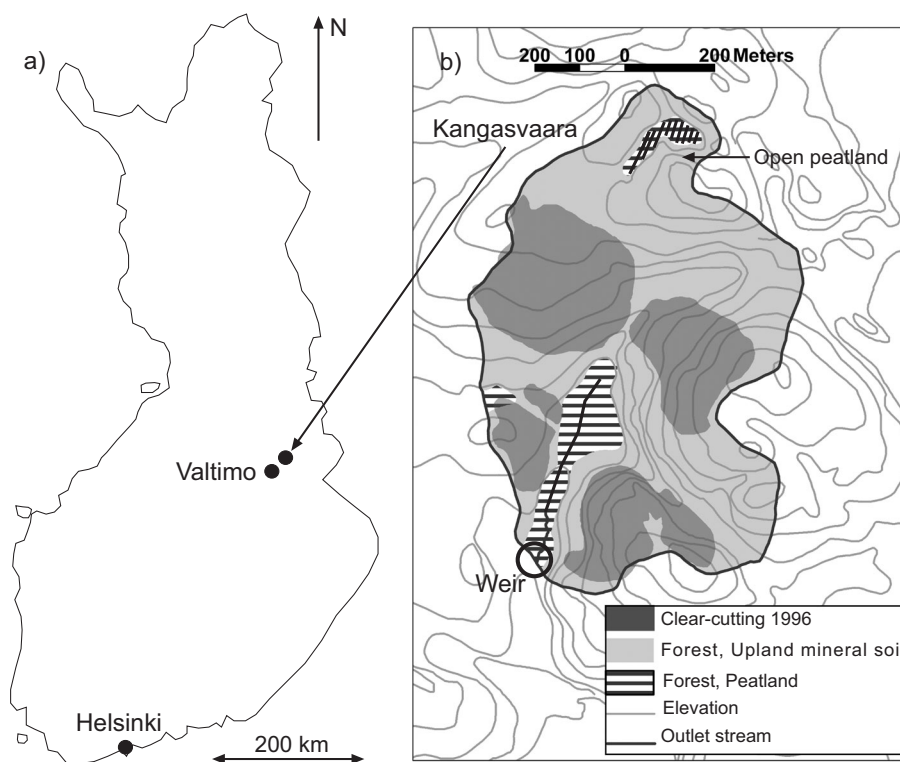


Fig. 1. (a) Location and (b) experimental setup of the Kangasvaara catchment in Eastern Finland.

to the Finnish classification (Cajander, 1949). The forests on the mineral soil are old-growth Norway spruce (*Picea abies* (L.) Karsten) mixed with Scots pine (*Pinus sylvestris* L.), silver birch (*Betula pubescens* Ehrh. and *B. pendula* Roth.) and European aspen (*Populus tremula* L.) (Finér *et al.*, 1997). The mean stem volume of the stands is 275 m³ ha⁻¹. The peatland area consists of pristine mesotrophic spruce-dominated mires.

In August–October 1996, five compartments totalling 19 ha of the upland mineral soil forest area were clear-cut and stems were removed (239 m³ ha⁻¹) (Fig. 1). Uncut buffer zones were left between the clear-cut area and the stream (Finér *et al.*, 2003). In August 1998, the soil was disc ploughed in strips to facilitate forest regeneration and in spring 1999 Scots pine seedlings were planted.

EXPERIMENTAL DATA USED

Several experimental datasets from the Kangasvaara catchment were used to construct and calibrate FEMMA. Hourly meteorological measurements of air temperature, relative humidity, global radiation, wind speed and precipitation since 1992 were combined with records from the nearest (*c.* 20 km, Valtimo) weather station operated by the Finnish Meteorological Institute (Koivusalo *et al.*, 2005).

Daily runoff measurements from a V-notch weir were available for five years before (1992–1996) and after clear-cutting (1997–2001) (Finér *et al.*, 1997). Concentrations of N were measured from water samples taken from the weir at two-week intervals during spring and autumn, and monthly in summer and winter. The N fractions were NH₄-N, combined nitrite-N and nitrate-N (referred to here as NO₃-N) and dissolved total N (DTN) (Finér *et al.*, 1997). DTN was analysed from unfiltered water samples. DON was calculated by subtracting the inorganic N fractions (NH₄-N and NO₃-N) from DTN. Monthly exports of N were calculated using a discharge-weighted method described by Rekolainen *et al.* (1991).

For the development and calibration of the decomposition sub-model included in FEMMA the results of litterbag experiments (Palviainen *et al.*, 2004) were used; these included the annual mass loss and N content of branch, leaf and fine root logging residues for three consecutive years after clear-cutting. For ground vegetation dynamics, the five-year dataset of Palviainen *et al.* (2005) was used.

Spatial data on elevation and soil depths were obtained by the Geological Survey of Finland using a Global Positioning System (GPS) and ground penetrating radar along a systematic network of transects at 70–120 m intervals. Elevation of the soil surface and depth of soil were calculated on a 10 × 10 m² grid using geo-statistical

interpolation. Soil hydraulic characteristics used in the model were taken from Koivusalo *et al.* (2005).

DESIGN AND PARAMETERISATION OF FEMMA

Overall structure

The FEMMA model combined and modified existing models with new ones (Fig. 2). Hydrological fluxes are described for canopy and snow accumulation and melt (Canopy and snow model, Koivusalo *et al.*, 2001) and water fluxes and storages along the hillslope (Characteristic profile model, Karvonen *et al.*, 1999). Tree-stand net photosynthesis and production of litter are described with FINNFOR (Kellomäki and Väisänen, 1997) and the decomposition of organic matter with ROMULN. Nitrification and denitrification are calculated with models presented in Jansson and Karlberg (2001). N uptake, transport with surface and ground waters, retention by soil and immobilisation in litter and logging residues are also simulated.

The catchment is described as a two-dimensional hillslope (Fig. 3), which is a longitudinal section extending from the water divide to the stream and from the soil surface down to the bedrock. The hillslope is divided into compartments distinguishing different site and soil types, stand characteristics or treatments in the catchment. Above-ground hydrological processes and forest dynamics are simulated separately for each compartment. For calculation of the processes in the soil, the hillslope is divided into vertical columns and the columns subdivided into horizontal layers. The transport of water and N between the layers and the columns is calculated. The hillslope is formed from a digital elevation model (DEM) to represent typical flowpaths of water inside the catchment (Kokkonen *et al.*, 2001). Water and associated N fluxes leaving the hillslope feed into a linear storage, which represents retention in a stream. A daily time step is used.

Two-dimensional description of the catchment

The surface boundary of the hillslope was calculated as the mean elevation difference between DEM pixels and its receiving stream pixel at a given distance range along the flowpath. For each distance range, the mean soil depth was determined from the interpolated soil depth grid. The number of pixels located at a given distance along the profile is used to calculate the relative width for the hillslope (Shreve, 1969). The location and proportion of forest, clear-cutting and buffer zone compartments along the hillslope were similarly assigned according the distribution of pixels (Fig. 3). The forest stand characteristics for forest and buffer zone compartments were determined according to field observations (Table 1). Peatland and mineral soil areas along

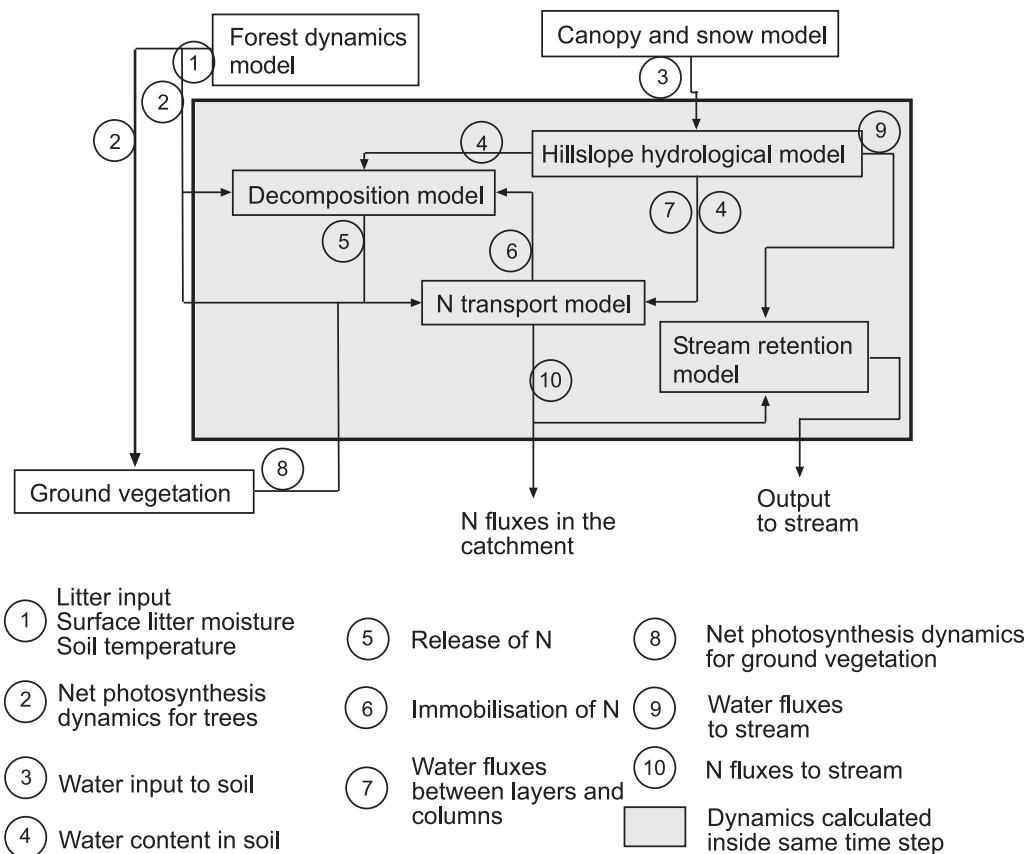


Fig. 2. Overall structure of the FEMMA model and the variables transferred between the sub-models. The sub-models inside the grey box are integrated into same calculation routine and computed inside one time step

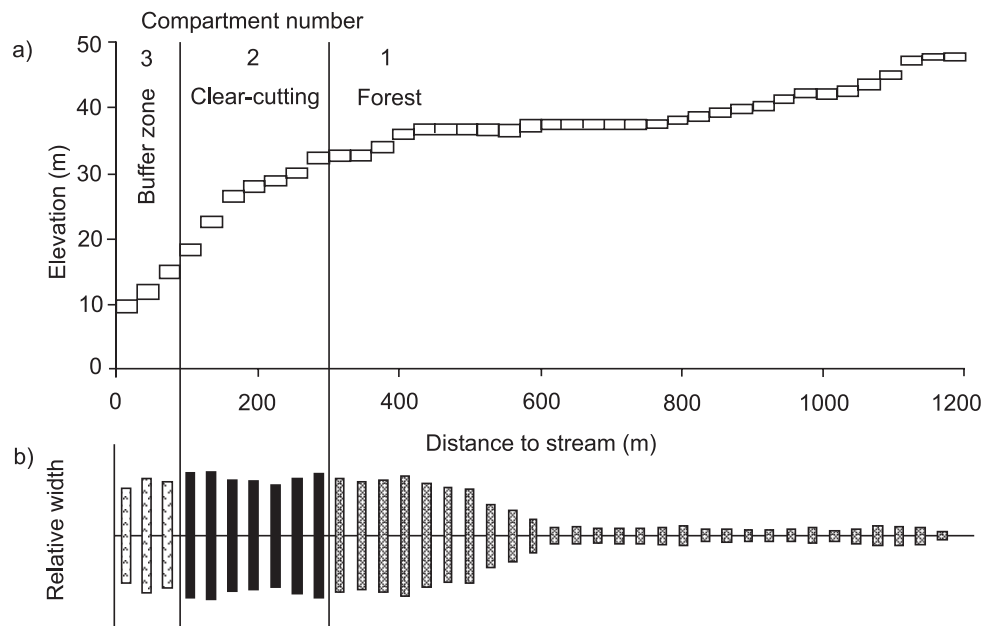


Fig. 3. (a) Elevation and (b) relative width of the hillslope representing the Kangasvaara catchment. The profile consists of 40 columns of 30 m length and depth to bedrock ranging from 0.9 to 2.1 m. Three compartments describing forest stand, site type and soil type are identified: 1 – mature Norway spruce on upland mineral soil, 2 – same as 1 but clear-cutting from October 1996, 3 – uncut buffer zone between the stream and the clear-cutting comprising of mature Norway spruce on upland mineral soil and peatland. The relative cover of each compartment type within the catchment was 0.51 for Compartment 1, 0.35 for Compartment 2 and 0.14 for Compartment 3.

Table 1. Forest stand characteristics in Kangasvaara catchment. These were used as input variables for the FINNFOR model applied for the forest and the buffer zone compartments.

Stratum	Tree species	Mean breast height diameter, cm	Mean height, m	Age, a	Stem density, stems ha ⁻¹
Dominant layer	<i>Picea abies</i>	21	18	130	800
Suppressed	<i>Picea abies</i>	10	9	100	800

the profile are also assigned based on the spatial distribution of these two soil types at a given distance from the stream. For Kangasvaara, the profile was discretised into 30 m long columns, which yielded a total of 40 columns. Further explanation of how the characteristic profile is determined is described by Koivusalo *et al.* (2005).

Above-ground hydrology

Above-ground hydrology is simulated in one dimension for each compartment present in the hillslope profile. The canopy routine simulates solar radiation, long-wave radiation and wind speed beneath the canopy and throughfall amount from meteorological variables characterising conditions above the canopy (Koivusalo and Kokkonen, 2002). Relative humidity and air temperature are assumed not to be affected by the canopy. The snow routine was based on the energy balance approach (Koivusalo *et al.*, 2001). Canopy and snow routine was run at an hourly time step. The parameterisation of the canopy and snow model for Kangasvaara has been described by in detail by Koivusalo *et al.* (2005).

Net photosynthesis and litter-fall production

Tree-stand net photosynthesis, litter production and soil temperature at different depths were calculated for each forest compartment using the FINNFOR-model (Kellomäki and Väisänen, 1997) at an hourly time-step. FINNFOR was parameterised for dominant and suppressed tree strata according to field measurements at Kangasvaara (Table 1). FINNFOR also produced estimates of annual litter-fall production and N contents for five fractions: branches, fine roots, foliage, roots and stems. To provide daily litter-fall inputs for the organic matter decomposition model, the annual litter-fall values was distributed for the period from September to October and for two weeks in May, which represented the litter-fall accumulated on snow. For other days litter-fall were considered to be zero.

Ground vegetation biomass and N pools dynamics were calculated from experimental data from Kangasvaara. Above-ground biomass and N concentrations of mosses, blueberry (*Vaccinium myrtillus* L.), lingonberry (*V. vitis-idaea* L.), grasses and herbs, and the roots of all the species,

have been determined before and after clear-cutting (Palviainen *et al.*, 2005). To represent the daily net photosynthesis of the ground vegetation, that of the tree-stand simulated with FINNFOR was scaled to match the ground vegetation biomass measurements. The annual litter production of the ground vegetation was estimated from the measured biomass according to biomass/litter-fall ratios presented by Mälkönen (1974). For the decomposition model, the ground vegetation litter-fall production estimates and their N contents were combined with the FINNFOR foliage and fine root litter production values. Daily litter-fall was then obtained by distributing the annual values to the period from September to October.

The ground vegetation dynamics were further modified to include the effect of site preparation on N dynamics. The area of soil where mineral soil was exposed or where the vegetation was covered with soil in disc ploughing was estimated to be 40% at Kangasvaara and the vegetation which died was transferred to the decomposition model. Vegetation on the area of soil remaining intact is assumed to develop and produce litter-fall.

Water fluxes along hillslope

Soil water movement down the hillslope and runoff generation were simulated with the characteristic profile model, CPM (Karvonen *et al.*, 1999; Koivusalo and Kokkonen, 2003). CPM takes input daily throughfall/snowmelt and potential transpiration values aggregated from the hourly output from the canopy and snow models. CPM is a quasi-two-dimensional model in the sense that vertical and lateral water fluxes were computed alternately. Vertical fluxes in all columns of the hillslope were computed using an approximation of the Richards equation with successive steady-state solutions of the pressure head distribution (Skaggs, 1980). Infiltration into a soil column is controlled by the available air-filled pore volume in the column. Water that does not infiltrate is transported downslope and infiltrates into the next column further down the slope if pore volume allows, or to the stream (surface runoff).

After the vertical fluxes and the resulting groundwater levels have been resolved, the lateral groundwater flows between vertical soil columns are computed according to

Darcy's law. Groundwater flow from the column next to the stream constitutes the baseflow component. When groundwater level in any column rises above the soil surface, water flows downslope as exfiltration. The sum of all runoff components (surface runoff, exfiltration and baseflow) is passed through a linear storage, which describes the delay of water flowing in the stream. Detailed parameterisation of CPM for Kangasvaara catchment is presented by Koivusalo *et al.* (2005).

Organic matter decomposition and N mineralisation

Organic matter decomposition was simulated with a routine modified from the ROMUL model (Chertov *et al.*, 2001). ROMUL was developed for forest soils having a raw humus (mor), moder or mull humus forms (Klinka *et al.*, 1981). Decomposition is modelled in three different stages: fresh litter material (L), complex humic substances with undecomposed organic debris (F) and humus material (H) (Chertov and Komarov, 1997).

In ROMUL, the release of N is remarkably lower than that of carbon, describing the high rate of N consumption by micro-organisms. However, ROMUL does not account for such N immobilisation by micro-organisms, which actually increases the absolute N content above the initial amount of N in the decaying litter cohorts (Berg and Söderström, 1979; Berg and Staaf, 1981; Staaf and Berg, 1982; Berg and Theander, 1984; Berg, 1988; Hasegawa and Takeda, 1996; Gebauer *et al.*, 2000; Palviainen *et al.*, 2004).

The original structure of ROMUL was modified such that the N dynamics would include immobilisation as described above but representation of the carbon dynamics was not changed. The modified ROMUL is referred to as ROMULN. The N immobilisation was modelled according to the following principles:

- (1) The dynamics of soil micro-organisms were modelled implicitly by including the immobilised N in the soil organic matter N pool.
- (2) Immobilisation occurs when the combined concentration of N in L and F pools falls below a critical concentration level, which is given as a parameter for each litter fraction (e.g. Kirschbaum and Paul, 2002), and when the soil temperature is above 0°C. The immobilisation N demand is calculated as N deficit with respect to critical concentration.
- (3) N for the immobilisation was extracted from the pool of dissolved N (e.g. Molina *et al.*, 1983; Li *et al.*, 1992; Kirschbaum and Paul, 2002). Immobilised N was taken to the F pool. The Mf parameter used in original ROMUL (Chertov *et al.*, 2001) to slow the release of N from F pool was set to 1.0.

Table 2. Parameter values in the ROMULN soil organic matter decomposition model

<i>Parameter</i>	<i>Unit</i>	<i>Value</i>	<i>Determined</i>
Organic matter content in mineral soil	% grav.	3.6	fixed <i>a priori</i>
Ash concentration of litter	% grav.	2	fixed <i>a priori</i>
Critical concentration in			
branches		0.5	calibrated
fine roots	% N	0.5	calibrated
foliage	from	1.9	calibrated
roots	dry mass	0.3	calibrated
stems		0.3	calibrated

ROMUL requires gravimetric water contents to simulate decomposition rates. The calculation of the gravimetric water contents from volumetric water contents and bulk densities is rather sensitive to the bulk density of the soil. To reduce this sensitivity and to ensure restriction of decomposition under wet conditions, an extra control was added to ROMULN. When the air-filled porosity (e_A , in volumetric %) in soil was lower than 15% of volume, the water content functions (denoted g1–g6 in Chertov *et al.*, 2001) were multiplied by $(1 - (15 - e_A)/15)$, on the basis of experimental results that soil respiratory activity connected with decomposition decreased when e_A fell below 10–20% of the soil volume (Glinski and Stepniewski, 1985).

ROMULN was integrated with the CPM and FemmaN (described later) at the top of each column along the hillslope. ROMULN receives the daily water contents in mineral topsoil from the CPM-model, daily moisture contents in the above-ground organic matter, soil temperature, litter-fall production and N inputs (for branches, fine roots, foliage, roots and stems) from the FINNFOR-model output, and for the ground vegetation, as described earlier, and microbiologically immobilised N from FemmaN-model (see Fig. 2). N released from organic matter computed by ROMULN is considered to be dissolved total N (DTN), and is subsequently used as input to the FemmaN-solute transport model. ROMULN was run at daily time step. Parameterisation of ROMULN for Kangasvaara is presented in Table 2.

Nitrogen fluxes along hillslope

The transport of different N fractions down the hillslope was calculated with a two-dimensional solute transport model, FemmaN. The modelled N fractions are $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, DON and DTN (the sum of the former three fractions). The time step used for FemmaN is one day.

N pools are calculated for each column and layer along the hillslope (Fig. 3). For forest, $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$ and DON deposition inputs were calculated from the observed mean concentrations for throughfall at Kangasvaara (Piirainen *et al.*, 1998; Piirainen *et al.*, 2002a) and from water input to the soil surface calculated with the Canopy and Snow-Model (Fig. 2). For the clear-cut area the concentration for the deposition was taken from the measured bulk precipitation data. The daily N input to soil arising from the decomposition of organic matter was received from ROMULN output. ROMULN DTN values were divided into $\text{NH}_4\text{-N}$ and DON fractions (Table 3); $\text{NO}_3\text{-N}$ was assumed not to be formed in decomposition. The majority of DTN was assumed to be $\text{NH}_4\text{-N}$ for mineral soil columns and DON for peatland columns (Hannam and Prescott, 2003; Potila and Sarjala, 2004).

The horizontal and vertical transport of N through the soil was calculated by the method of Jansson and Karlberg (2001). Retention of $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ was computed with layer-specific adsorption coefficients taken from Jansson

and Karlberg (2001) (Table 3). DON can be retained in both mineral and peat soils (Qualls, 2000) but adsorption coefficients for DON are not available and values were calibrated. Nitrification and denitrification fluxes were calculated as a function of soil $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ concentrations and of temperature and moisture for each soil layer (Jansson and Karlberg, 2001). In forest soils the rate of nitrification is often reduced by low pH (e.g. Persson and Wirén, 1995; Persson *et al.*, 2000). The reduction factor for nitrification (npH) was calculated using Eqn. 1 (Wu and McGechan, 1998):

$$\text{npH} = (\text{pH} - \text{pHmin}) / (\text{pHmax} - \text{pHmin}) \quad (1)$$

where pH is the actual pH value of the soil, pHmax is the pH value at which nitrification is not affected by the acidity and pHmin is the value at which nitrification is zero. The soil pH-values are 4.0 and 4.2 for the organic and mineral soil layers respectively (Piirainen *et al.*, 2002b). A value of 3.8 for pHmin and 5.4 for pHmax was based on data presented by Persson *et al.* (2000).

Gaseous N_2O -N fluxes

N_2O -N can be produced by nitrification, although it is usually considered to be produced during denitrification (Ambus, 1998). However, in laboratory experiments, Martikainen (1985) has shown that nitrification in acid forest soil may release substantial amounts of N_2O -N. When soil pH (in water solution) was less than 4.1, 20% of the oxidised $\text{NH}_4\text{-N}$ was converted into N_2O -N. The percentage ($P_{\text{N}_2\text{O-N}}$) of the oxidised $\text{NH}_4\text{-N}$ becoming N_2O -N instead of $\text{NO}_3\text{-N}$ was estimated using Eqn. 2 (Martikainen, 1985):

$$P_{\text{N}_2\text{O-N}} = \text{Min} (20.0, 10^{(-2.387 \text{pH} + 11.184)}). \quad (2)$$

Plant N uptake

The N demand of the uptake by trees was calculated from the modelled daily net photosynthesis values by dividing it by the nutrient use efficiency, NUE (Gourley *et al.*, 1994). The NUE (Table 3) was calculated from forest biomass and nutrient content data presented by Finér *et al.* (2003) for Kangasvaara. The NUE for the ground vegetation was calculated for Kangasvaara from measured biomass and N contents (Palviainen *et al.*, 2005).

Trees and ground vegetation take up N from the whole rooting zone (depth 30 cm), whereas the microbes take up N only from the top soil layer (10 cm). The plants and microbes preferably take up inorganic forms of N ($\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$) but, if the inorganic N pool is insufficient, then DON is also taken up. DON contains amino acids, which for arctic plants may account for 10–82% of the total N

Table 3. Parameters for FemmaN –model.

Parameter	Unit	Value	Determined
Nutrient use ratio (NUE)			
trees	kgC/kgN	125	fixed <i>a priori</i>
ground vegetation	kgC/kgN	60	fixed <i>a priori</i>
Nitrification model			
npH	[-]	0.125	fixed <i>a priori</i>
nrate	[-]	0.2	calibrated
nrate (clear-cut compartment)	[-]	0.1	calibrated
Fractioning of released DTN			
mineral soil			
$\text{NH}_4\text{-N}$	%	95	calibrated
DON	%	5	calibrated
peat soil			
$\text{NH}_4\text{-N}$	%	50	calibrated
DON	%	50	calibrated
Adsorption coefficients			
mineral soil			
$\text{NH}_4\text{-N}$	[-]	0.99	fixed <i>a priori</i>
$\text{NO}_3\text{-N}$	[-]	0.0	fixed <i>a priori</i>
DON	[-]	0.99	calibrated
peat soil			
$\text{NH}_4\text{-N}$	[-]	0.99	fixed <i>a priori</i>
$\text{NO}_3\text{-N}$	[-]	0.0	fixed <i>a priori</i>
DON	[-]	0.75	calibrated

uptake (Kielland, 1994). Näsholm *et al.* (1998) has shown that certain boreal forest tree species (e.g. *P. sylvestris*, *P. abies*), dwarf shrubs (e.g. *V. myrtillus*) and grass (e.g. *Deschampsia flexuosa*) also take up amino acids. However, amino acids comprise only some 1–1.5% of the DON in soil water (Hannam and Prescott, 2003); therefore, the DON uptake was restricted to a maximum of 1% of the current DON storage. The vertical distribution of N uptake in a column was taken to correspond to the relative abundance of N in each soil layer within the rooting zone.

CALIBRATION

Parameters for the hydrological model were calibrated to Kangasvaara (Koivusalo *et al.*, 2005). In calibrating the FemmaN -model, all the measured N concentrations in the outlet stream were used and so the model could not be validated against streamwater chemistry. Those parameter values presented in literature were fixed *a priori*. The calibrated parameters included *nrate* parameter (nitrification in optimal conditions) in the nitrification model for the clear-cutting area, DON adsorption parameter for mineral and peat soils and fractioning parameter for DTN released in the decomposition. Parameters for the critical concentration in ROMULN-model were calibrated against litterbag experiments presented in Palviainen *et al.* (2004).

NITROGEN FLUX SIMULATIONS

Simulations were performed for the period starting from January 1, 1992, to December 31, 2001, using two scenarios, one represented the reality, in which clear-cutting (October 1996) and site preparation (August 1998) in the catchment had been carried out. The other scenario was a control without any treatments. The difference represented the effect of the clear-cutting and site preparation. Daily time series of runoff and concentrations and export of NO₃-N, NH₄-N and DON in the stream were computed and compared against field measurements.

The N mass balance for the forested, clear-cut and buffer zone compartments in the catchment (denoted as 1–3 in Fig. 3) were also computed. For each compartment, the influxes were the ground- and surface-water influxes from upslope, and the deposition and decomposition N flux; the outfluxes were the ground- and surface-water outflux downslope, and tree and ground vegetation uptake, microbial immobilisation, denitrification and emissions of N₂O-N. The change in the adsorption storage of the soil was calculated. Each flux was calculated for NH₄-N, NO₃-N and for DON.

The goodness of fit between modelled and measured values was assessed in terms of the mean absolute error

(ME):

$$ME = \frac{\sum_i |x_i - m_i|}{n} \quad (3)$$

where x_i is the simulated value at time i , m_i is the measured value, and n is number of the measurements in the time series. The goodness of fit was also assessed by correlation analysis of the measured and calculated time series.

Results

OUTPUT TO STREAM

Nitrogen concentrations

The FEMMA-model enabled reproduction of the seasonal and treatment-induced changes in streamwater N concentrations (Fig. 4). Calculated DTN concentrations were higher in summer than in winter, following the pattern of the measured concentrations (Fig. 4). For DTN, the ME was slightly less than half the measured mean concentration (Table 4). DON was the dominant form of N in streamwater, accounting for 90% of DTN on average, according to both calculations and measurements. Thus, the temporal pattern in DON concentrations and the goodness of fit of the model were rather similar to that for DTN. The simulations also reproduced the seasonal pattern in NO₃-N concentrations; the annual maxima occurred before the spring flood and decreased thereafter. Measured concentrations of NH₄-N in the stream showed no distinctive seasonal variation and were low, being only slightly above detection limit concentrations throughout the year. There was no correlation between the measured and calculated NH₄-N concentrations.

The effects of clear-cutting in August–October 1996 indicated no change in the seasonal pattern of N concentrations until autumn 1999, when NO₃-N concentrations began to increase. This was in accordance with the measurements (Fig. 4). The increase in NO₃-N concentrations was the most distinctive effect of clear-cutting. However, DON and NH₄-N concentrations also increased during 2000. The increase in DON concentrations was slightly less than for NO₃-N but the relative increase was rather small.

Nitrogen export to the stream

The FEMMA-model mimicked the monthly and annual export of N to the stream with reasonable accuracy (Fig. 5, Table 4), compared with measurements (Table 5). N export values induced by clear-cutting increased in spring 1997 but remained low until spring 2000. The greatest N export was calculated for 2000 and, during 2001, N export started

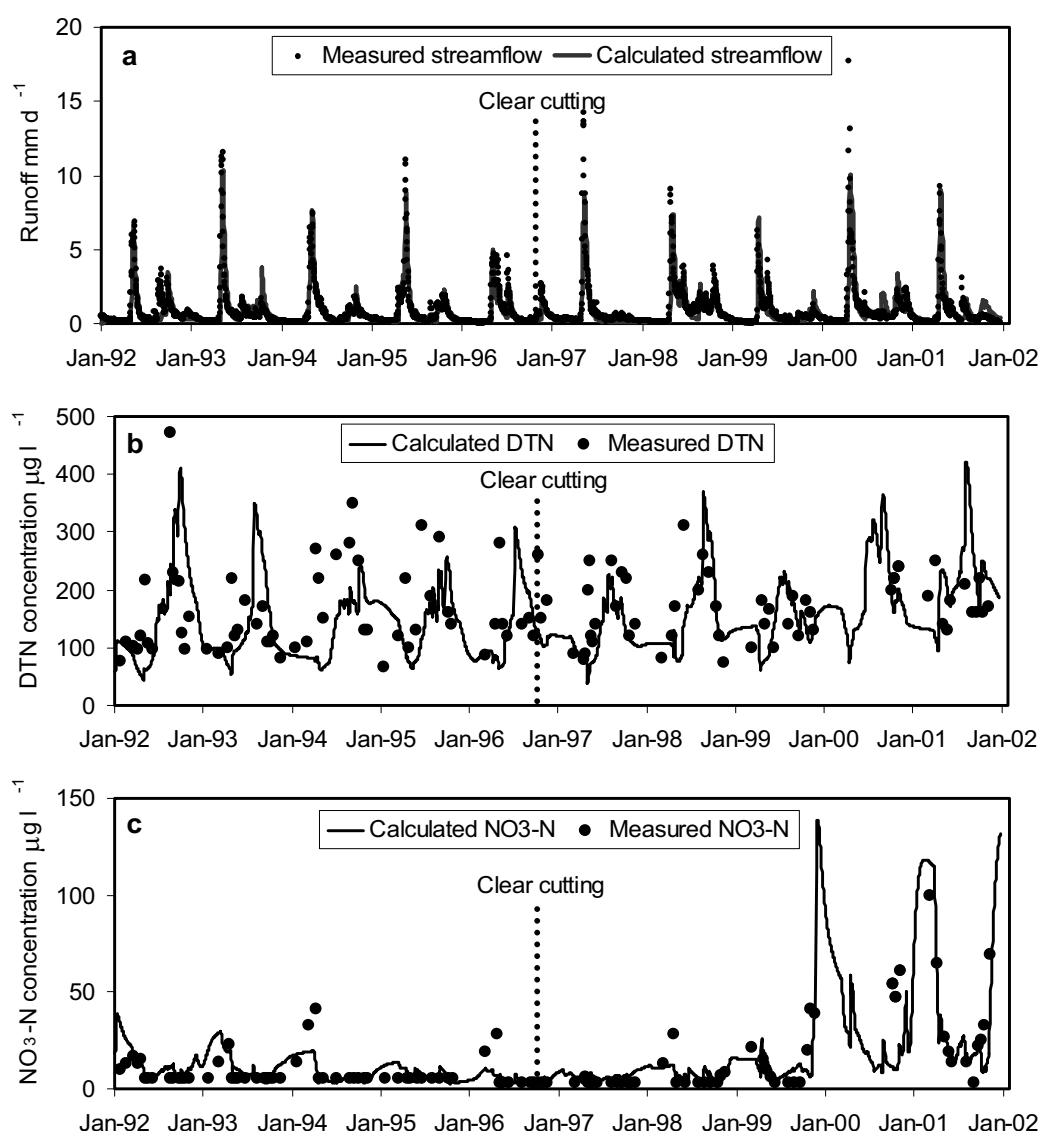


Fig. 4. Stream runoff (mm d⁻¹) (a), measured and simulated concentrations of dissolved total nitrogen (DTN) (b) and nitrate-N (NO₃-N) (c) in Kangasvaara outlet stream (µg N l⁻¹). Clear-cutting in the catchment was performed during August–October 1996.

Table 4. Mean of observed stream N concentrations and exports (monthly values, 1992–2001) and mean error (ME) of modelled results (calculated with Eqn. 3) and Pearson coefficients (r) between the observed and calculated values (n = 110).

	Concentration, mg l ⁻¹			Monthly N export to stream, kg ha ⁻¹		
	Mean	ME	r	Mean	ME	r
DTN	166	75	0.27**	0.043	0.025	0.63**
NH ₄ -N	3	3	-0.10	0.001	0.001	0.32**
NO ₃ -N	13	4	0.69**	0.003	0.002	0.57**
DON	151	74	0.26**	0.040	0.024	0.60**

**Significant at P < 0.01

to decrease. According to the simulations, a substantial part of the annual N export took place during springtime. In the control scenario, 45% of the annual export of NO₃-N and 30% of the annual export of DON occurred between April 15 and May 31, on average. Clear-cutting increased the episodic behaviour of the DON export, 37% of DON being exported between April 15 and May 31.

NITROGEN FLUXES WITHIN THE CATCHMENT

Control scenario

The mass balance of N for each soil column along the hillslope was calculated and the results were lumped for each of the three compartments within the catchment (Table

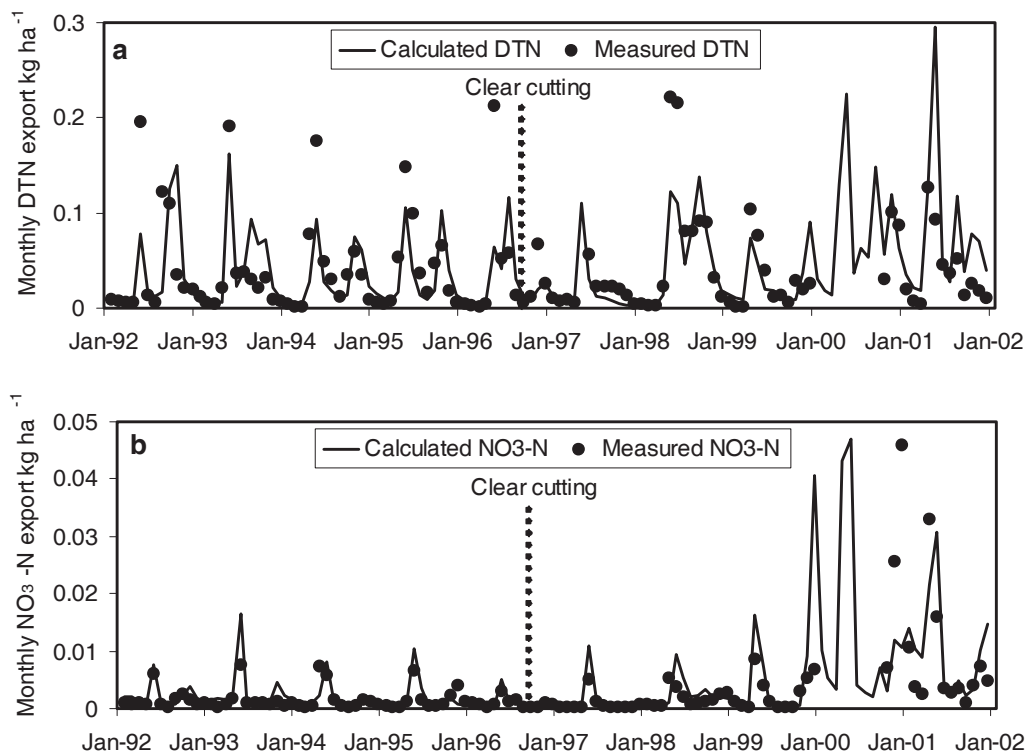


Fig. 5. Observed and simulated exports ($\text{kg N ha}^{-1} \text{ month}^{-1}$) of total dissolved nitrogen (DTN) (a) and nitrate ($\text{NO}_3\text{-N}$) (b) in Kangasvaara outlet stream. Clear-cutting in the catchment was performed during August–October 1996.

Table 5. Mean annual N export ($\text{kg N ha}^{-1} \text{ a}^{-1}$) from the Kangasvaara catchment according to simulations and measurements.

	DTN	$\text{NH}_4\text{-N}$	$\text{NO}_3\text{-N}$	DON
BEFORE CLEAR-CUTTING				
simulated	0.424	0.001	0.018	0.399
measured	0.484	0.010	0.024	0.456
AFTER CLEAR-CUTTING				
simulated	0.509	0.005	0.067	0.436
measured	0.555	0.007	0.055	0.493

6). The most important influx of N was the decomposition of organic matter, contributing $51\text{--}54 \text{ kg DTN ha}^{-1} \text{ a}^{-1}$; the deposition input was *c.* $5 \text{ kg DTN ha}^{-1} \text{ a}^{-1}$ (Table 6). In the control scenario, the N influx with the lateral flow of surface and groundwater was less than $0.4 \text{ kg DTN ha}^{-1} \text{ a}^{-1}$, the amount increasing slightly downslope and in convergent locations on the hillslope. Most of the transported DTN was carried by lateral flow of surface waters. Uptake of N by trees, ground vegetation and immobilisation to microbes were the main outfluxes of N within the catchment: tree uptake was $20\text{--}22 \text{ kg DTN ha}^{-1} \text{ a}^{-1}$, ground vegetation uptake

was $14\text{--}18 \text{ kg DTN ha}^{-1} \text{ a}^{-1}$ and microbial immobilisation was $18\text{--}19 \text{ kg DTN ha}^{-1} \text{ a}^{-1}$. The soil N storage changed little during the simulation period in the control scenario and only a small proportion of N was exported to the stream ($0.42 \text{ kg DTN ha}^{-1} \text{ a}^{-1}$).

Clear-cutting and buffer zone

Upon clear-cutting, the N fluxes changed in the clear-cut area and buffer zone. The amount of dead organic matter increased in the clear-cut area resulting in increased decomposition compared with the control scenario (Table 6). The annual mass balances show that decomposition at the end of the simulation period was still higher than before the cutting (Table 7). N uptake by trees decreased but the microbial immobilisation increased to $71 \text{ kg DTN ha}^{-1} \text{ a}^{-1}$ in the year following the clear-cutting. However, by the third year (1999) after harvesting the microbial immobilisation of N decreased to less than $1 \text{ kg ha}^{-1} \text{ a}^{-1}$. Uptake by ground vegetation also was lower in the first two years after clear-cutting but, in 1999, it was similar to that before the clear-cutting (Table 7). Adsorption storage in soil was the main sink of N from 1998 onwards. After clear-cutting, nitrification increased year by year throughout the simulation period.

Table 6. Modelled mean annual mass balance of total dissolved nitrogen (DTN, kg ha⁻¹ a⁻¹) for the three modelled compartments under clear-cutting and no cutting scenarios. Calculations are done for the period from October 1996 to December 2001. Compartment 1 is next to the water divide and Compartment 3 is next to the outlet stream. Compartment 1 is mature Norway spruce forest on upland mineral soil, Compartment 2 is similar to Compartment 1 until clear-cutting in October 1996, Compartment 3 is a buffer zone with mature Norway spruce forest and partly on upland mineral soil and partly on peat soil. Note that fluxes are presented ha⁻¹ of the studied compartment and scaled according to the relative width of the hillslope, thus the outflux from the upslope range does not necessarily equal to influx downslope range in the table. Negative adsorption refers to decrease in adsorption storage.

Flux [kg N ha ⁻¹ a ⁻¹]	Compartment 1			Compartment 2			Compartment 3		
	SCENARIO			SCENARIO			SCENARIO		
	Cut	Control	Difference	Cut	Control	Difference	Cut	Control	Difference
INFLUXES									
Deposition	5.03	5.03	0.00	6.05	5.03	1.02	5.03	5.03	0.00
Decomposition	51.10	51.10	0.00	74.55	53.70	20.85	51.15	50.81	0.34
Surface water	0.00	0.00	0.00	0.07	0.07	0.00	15.57	0.29	15.28
Groundwater	0.00	0.00	0.00	0.01	0.01	0.00	3.19	0.09	3.10
OUTFLUXES									
Tree uptake	20.34	20.34	0.00	2.27	21.59	-19.32	28.46	21.46	7.00
Ground vegetation uptake	16.62	16.62	0.00	17.30	17.59	-0.29	18.94	14.28	4.66
Immobilisation	19.24	19.24	0.00	19.23	19.61	-0.38	19.23	17.54	1.69
Denitrification	< 0.01	< 0.01	0.00	< 0.01	< 0.01	0.00	<0.01	<0.01	0.00
N ₂ O-N emission									
in nitrification	0.01	0.01	0.00	0.59	0.01	0.58	0.09	0.01	0.08
Surface water	<0.01	<0.01	0.00	6.01	0.11	5.90	3.99	2.81	1.18
Groundwater	<0.01	<0.01	0.00	0.52	0.03	0.49	0.52	0.42	0.10
Δ ADSORPTION									
storage in soil	-0.14	-0.14	0.00	34.51	-0.14	34.65	3.70	-0.33	4.03
OTHER FLUXES									
Nitrification	0.04	0.04	0.00	2.94	0.04	2.90	0.47	0.03	0.44

Clear-cutting increased the transport of N with groundwater and surface water flows. The absolute amount of DTN transported from the clear-cut area (Compartment 2) to the buffer zone (Compartment 3) increased by 140 kg a⁻¹ (Table 8). However, the transport of DTN from the buffer zone to the stream increased by only 10 kg a⁻¹. According to the simulation, 76% of the DTN exported from the clear-cut area was retained in the buffer zone. Mass balance calculations showed an increase in tree and ground vegetation uptake, microbial immobilisation and soil adsorption storage within the buffer zone (Table 6).

Discussion

FLUXES EXCLUDED FROM THE MODEL AND THE LIMITATIONS OF THE MODELLING FRAMEWORK

The export of N from a forested catchment to the outlet stream is affected by many simultaneous interactive

processes that are controlled by physical, chemical and biological factors. While the main N fluxes have been included in the model, explicit calculation of biological N fixation has been omitted. In boreal forests, biological N₂ fixation associated with feather moss (*Pleurozium schreberi* (Bird.) Mitt) can be 1.5–2 kg N ha⁻¹ a⁻¹ (DaLuca *et al.*, 2002) and the asymbiotic N fixation < 1 kg N ha⁻¹ a⁻¹ (Hendrickson, 1990). In FEMMA, N fixed by the mosses was included implicitly in the moss litter input but the asymbiotic N fixation flux was omitted.

The structure of FEMMA sets some limitations to its applications. In representing the catchment as a single hillslope some spatial information is lost. The hydrological model was designed for sites with shallow soils with an impermeable lower boundary (Skaggs, 1980) and so is applicable to the Kangasvaara site. Furthermore, preferential flow in soil macropores, which can be significant in nutrient transport (Kareinen *et al.*, 1998), is not accounted for explicitly in the model.

Table 7. Annual mass balance of DTN for the clear-cut area before and after harvest.

Flux [kg N ha ⁻¹ a ⁻¹]	Before	1997	1998	1999	2000	2001
INFLUXES						
Deposition	5.03	4.90	6.84	5.23	7.75	6.12
Decomposition	57.57	87.82	70.80	89.66	65.31	66.53
Surface water	0.12	0.01	0.09	0.06	0.10	0.07
Groundwater	0.02	0.06	0.01	0.01	0.01	0.01
OUTFLUXES						
Tree uptake	26.46	0.08	0.25	2.29	3.46	5.47
Ground vegetation uptake	19.40	16.06	12.12	19.39	19.31	21.31
Immobilisation	14.54	70.54	18.21	0.57	2.63	3.67
Denitrification	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
N ₂ O-N emission						
in nitrification	0.02	0.03	0.16	0.70	0.93	1.19
Surface water	0.18	0.10	1.62	3.45	14.51	10.98
Groundwater	0.01	0.03	0.41	1.04	2.33	2.42
Δ ADSORPTION						
storage in soil	0.10	5.94	44.91	67.53	29.99	27.67
OTHER FLUXES						
Nitrification	0.10	0.14	0.72	3.47	4.65	5.93

Table 8. Mean annual simulated N transport (kg DTN ha⁻¹ a⁻¹) from between the three compartments for the period from October 1996 to December 2001. For compartment descriptions, see Table 6.

Flux [kg N a ⁻¹] N fraction	Compartment 1 → Compartment 2			Compartment 2 → Compartment 3			Compartment 3 → Stream		
	SCENARIO			SCENARIO			SCENARIO		
	Cut	Control	Difference	Cut	Control	Difference	Cut	Control	Difference
DTN	1.64	1.64	0.00	143.34	2.90	140.44	34.51	24.72	9.79
NH ₄ -N	0.06	0.06	0.00	105.54	0.15	105.39	0.18	0.05	0.13
NO ₃ -N	0.92	0.92	0.00	20.72	1.62	19.10	4.50	1.30	3.20
DON	0.66	0.66	0.00	17.08	1.13	15.95	29.82	23.38	6.44

RESULTS IN THE CONTEXT OF PREVIOUS EXPERIMENTS

The model has been applied to a catchment in which the forest growth is N-limited and deposition low; N is, thus, tightly cycled and export to the stream is at the lower end of the range reported in other studies (Lepistö, 1995; Moldan and Wright, 1998b; Ahtiainen and Huttunen, 1999; Mattsson *et al.*, 2003; Åström *et al.*, 2003; Akselsson *et al.*, 2004).

One of the main goals of this study was to simulate N fluxes down a representative hillslope that extended from the water divide to the stream. In Sweden, Jacks and Norrström (2004) have studied N transport along transects

from both a clear-cut area on upland mineral soil through a riparian buffer zone to a stream as well as from a pristine boreal coniferous forest draining to a stream. The clear-cut area in their study was 73% of the catchment and the buffer zone 9% (corresponding values in the present study catchment were 35% and 14%). They estimated the N export using runoff data and N concentrations in water collected with piezometers installed along the transects. The N export from the clear-cut hillslope to the buffer zone was 41 kg N ha⁻¹ a⁻¹ and from the uncut hillslope 1.6 kg N ha⁻¹ a⁻¹ indicating that clear-cutting had caused a 26-fold increase in N exports to the buffer zone. According to their calculations, 73% of the N exported from the clear-cut area

was retained in the buffer zone during the one-year study period. In the present Kangasvaara simulations, the mean annual outflux from the clear-cut area was $6.48 \text{ kg N ha}^{-1} \text{ a}^{-1}$, while the outflux from the control scenario was $0.21 \text{ kg N ha}^{-1} \text{ a}^{-1}$ (see Table 6). Thus, clear-cutting had increased the N export to the buffer zone 31-fold and some 76% of the DTN was retained in the buffer zone (Table 8). Considering the differences between the sites, the uncertainties in the modelling and measurements, the results of these two studies are remarkably similar.

According to the simulation, the release of N associated with decomposition was the main source of N into the soil-water system. Experimental quantification of N release from organic material is rather problematic because several processes affect the net N release, including mineralisation, nitrification, denitrification, microbe assimilation and uptake (Persson *et al.*, 2000). Comparison of experimental N release studies is not always straightforward because measurement methods often differ (StAAF and Berg, 1982; Berg, 1988; Hasegawa and Takeda, 1996; Devito *et al.*, 1999; Andersson *et al.*, 2002; Bengtsson *et al.*, 2003; Vestgarden *et al.*, 2003). The ROMULN model was calibrated against litterbag results from the catchment studied using logging residues (Palviainen *et al.*, 2004). Critical concentration parameters (Table 2) in the model were adjusted to replicate the measured N contents in foliar, branch and fine root logging residues for three consecutive years after the clear-cutting. The changes made to the ROMUL model related mainly to the N dynamics of the F pool, in which fractions of an organic matter cohort stay for a maximum of a few years; the dynamics of the old organic matter in H succession phase, were not changed.

In the present simulations, the mean annual net N release (calculated as decomposition – immobilisation) was 31–34 $\text{kg N ha}^{-1} \text{ a}^{-1}$ in the control scenario and 17 $\text{kg N ha}^{-1} \text{ a}^{-1}$ during the first year after clear-cutting (Tables 6 and 7). The maximum simulated net annual mineralisation, 89 $\text{kg N ha}^{-1} \text{ a}^{-1}$, occurred three years after clear-cutting. These results fall within the range reported for Scandinavian forest soils, 0–110 $\text{kg N ha}^{-1} \text{ a}^{-1}$ (Persson and Wirén, 1995; Persson *et al.*, 2000; Kjønaas *et al.*, 1998; Andersson *et al.*, 2002).

Kjønaas *et al.* (1998) studied the effects of N addition on N cycling in three Swedish catchments of mature Norway spruce stands; the addition of N increased the assimilation by soil and microbes to 83 $\text{kg N ha}^{-1} \text{ a}^{-1}$ and net mineralisation to 75.5 $\text{kg N ha}^{-1} \text{ a}^{-1}$. In the present study, clear-cutting increased especially decomposition, immobilisation and soil storage; during the first year after the clear-cutting, 71 $\text{kg N ha}^{-1} \text{ a}^{-1}$ was assimilated by the microbe populations associated with the logging residues (Table 7), in agreement with the results of Kjønaas *et al.* (1998).

For Scandinavian forest soils, reported nitrification rates vary from 0–76 $\text{kg N ha}^{-1} \text{ a}^{-1}$ (Persson and Wiren, 1995; Kjønaas *et al.*, 1998; Persson *et al.*, 2000; Andersson *et al.*, 2002; Vestgarden *et al.*, 2003). In pristine conditions, the rate of nitrification is rather low. In the present study, nitrification under pristine conditions was 0.03–0.04 $\text{kg N ha}^{-1} \text{ a}^{-1}$; after clear-cutting it increased to a maximum of 5.93 $\text{kg N ha}^{-1} \text{ a}^{-1}$ in 2001 (Tables 6 and 7). However, these values should be viewed with caution, because nitrification parameter (*nrate*) was calibrated against measured nitrate concentrations in the stream. Soil pH in the topmost layer was 4.0, the critical value at which nitrification becomes low (Persson and Wirén, 1995). However, clear-cutting increases soil pH and available $\text{NH}_4\text{-N}$ such that nitrification may be initiated (Smolander *et al.*, 1998; Paavolainen and Smolander, 1998).

Nitrification can lead to the production of gaseous $\text{N}_2\text{O-N}$, although this is usually a minor source of $\text{N}_2\text{O-N}$ from forest soils compared to denitrification (Ambus, 1998). No values were found in the literature for $\text{N}_2\text{O-N}$ production from clear-cut areas but, according to the simulations, the flux of $\text{N}_2\text{O-N}$ from the clear-cut area was 0.59 $\text{kg N ha}^{-1} \text{ a}^{-1}$ (Table 6). For pristine peatlands and unfertilised coniferous forests on upland soils, $\text{N}_2\text{O-N}$ emissions are very low, a few tens of grams $\text{N ha}^{-1} \text{ a}^{-1}$ (Matson *et al.*, 1992; Martikainen *et al.*, 1993). $\text{N}_2\text{O-N}$ emissions from drained forested peatlands are higher (1.2–5.2 $\text{kg N ha}^{-1} \text{ a}^{-1}$) because drainage enhances the decomposition of peat and increases the amount of dissolved $\text{NH}_4\text{-N}$ in the soil (Martikainen *et al.*, 1993; Regina *et al.*, 1998a; Regina *et al.*, 1998b; Maljanen *et al.*, 2003). Fertilisation or deposition also increase the amounts of dissolved $\text{NH}_4\text{-N}$ in the soil, thereby enhancing the gaseous N emissions. However, effects on well drained mineral soils are smaller than on drained peatlands (Klemedtsson *et al.*, 1997). That could indicate that gaseous N emissions would increase after clear-cutting but not to the level at drained forested peatlands. $\text{N}_2\text{O-N}$ fluxes have been estimated simply as a linear function of nitrification although they may be affected by many factors, such as water content and temperature, freezing and thawing cycles and soil pH. The factors controlling gaseous fluxes at low temperature are still poorly known (Teepe *et al.*, 2000; Öquist *et al.*, 2004).

CHALLENGES FOR THE MODEL AND FUTURE PROSPECTS

The FEMMA model succeeded in simulating stream runoff and the annual export and seasonal dynamics of N concentration in the stream water. As with many models working at the daily time-step, the reproduction of responses

to individual events is not very good. Moreover, a 'correct' N export may reflect a combination of incorrectly modelled N fluxes within the catchment. Simulated N fluxes within the catchment were high compared with N export in stream water. The cycling of N between litter, microbes, soil and vegetation was almost closed, the measured annual export of DTN was only about 1% of the calculated annual release of N by decomposition and the exports of $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ into the stream were respectively only 1.6 and 0.8% of the measured deposition loads. The closed nature of the N cycle makes the experimental and modelling work challenging because errors in mass balance components can result in a big relative response to the observed and modelled output.

The approach used in FEMMA fills the gap between point-scale and large-scale models by simulating fluxes of water and N down-slope through different land-use types and including DON. Exploiting these new properties enable adjusting forest management practices inside the catchment in a way that minimises the N export. Some of these studies could be done with fully distributed three-dimensional models that have been developed (MIKE SHE, Boegh *et al.*, 2004), but applications to forested environments are rare. Previous modelling of N leaching to streams has concentrated on inorganic forms but FEMMA also simulates DON fluxes.

FEMMA should be further developed towards more generic applicability of the model. This requires testing the model against independent datasets from paired catchment studies conducted under different deposition, land-use and forest and soil types. Inclusion of phosphorous and acidifying ions into the model would widen the user group of the model. On the other hand, experimental element transport data along the water pathway on the hillslope is still required to test the validity of the results within the catchment.

Conclusions

The key results in the present work were:

- A new model enabling the down-slope routing of water and N through different land-use types and the inclusion of DON.
- The model simulation showed that the most important sinks of N after the clear-cutting were immobilisation by the soil microbes, uptake by ground vegetation and sorption to soil.
- Simulations suggested also that the buffer zone retained a substantial part of the N flux before reaching the outlet stream.

Currently, FEMMA can be applied only to experimental sites having data for calibration and so cannot yet be used as a decision support system in operational environment management. However, such applications identify which processes are particularly important in affecting N export and which combinations of forest management options, site type, soil type, catchment dimensions and climate risk high N loading to streams. This information can then be applied in prescribing practical forest management operations.

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